

**Effects of Land Cover on Aquatic Communities and Food
Webs: A Study of Second-Order Streams in Southeastern
Michigan.**

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Abstract

Lotic systems in many regions of the country have experienced habitat degradation and biodiversity loss from impacts brought about from agricultural activity and urbanization. Southeastern Michigan is no exception, as agriculture in the River Raisin watershed and continued expansion of suburban Detroit in the Huron River watershed threaten both systems. To further understand the ecological impact of land use on Midwestern streams I created and compared food webs for nine different second-order tributaries distributed equally across three generalized land use categories including; developed, agricultural, and undeveloped sub-basins of the Huron River and River Raisin. Fish diets (n=410) were analyzed to create the food webs, and weighted quantitative metrics were used to identify differences in fish-invertebrate interactions across streams with differing land use. Although undeveloped streams had higher diversity and less habitat degradation no significant differences were found in weighted quantitative metrics across the three stream categories. Decapoda, terrestrial Hymenoptera, and Chironomidae were the primary prey taxa in all stream categories. Decapods accounted for the majority of biomass consumed and the pattern of their consumption strongly influenced metric scores. Metric values were not significantly related to land use in the sub-basins or local habitat assessment scores. Weighted quantitative metrics may not be applicable in understanding how anthropogenic land use influences aquatic food webs where there is a dominant, tolerant prey taxa.

Introduction

Urban and agricultural development are common responses to population growth, and with development comes a variety of ecological impacts. Learning how anthropogenic land use affects natural systems may help mitigate and minimize the negative consequences of changing land cover. Altering landscapes affects not only terrestrial ecosystems but aquatic, ones as well (Allan 2004). It is well known that anthropogenic land use can

have severe impacts to streams and their communities, but how it alters interactions within these communities has not been well documented.

Over the last 30 years ecologists have pursued a variety of approaches to describing interactions within food webs (Winemiller 1990). Food webs from numerous terrestrial ecosystems have been reported, yet development of descriptors to present quantified information in an ecologically valuable manner has been challenging. Despite the expansion of food web research, much of this work has focused on identifying universal properties and community organization of food webs (Pimm 1979, Cohen and Briand 1984, Pimm et al. 1991). Such concepts have not gone without controversy. Paine (1988) criticized the quality of data used to derive some food web properties, and argued that many properties were too arbitrary to carry any biological significance. After completion of a tropical fish food web study, Winemiller (1990) concluded that past research overly stressed the importance of food web complexity and that there was a lack of emphasis on interaction strength in existing food web descriptors. Winemiller (1990) also found that because these descriptors were intended to identify natural patterns in community structure they lacked the power to detect community reactions to environmental changes. Further food web studies found irregularities in food web properties that were thought to be constant (Havens 1993, Martinez 1993).

In light of the criticism of early food web descriptors Bersier et al. (2002) presented a set of weighted quantitative descriptors. By using weighted quantitative descriptors they were able to account for inequality in links between organisms. Such descriptors advance over previous works by taking into account that some organisms may be very highly linked but only represent a small fraction of biomass flow in a community. Bersier et al. (2002) found weighted descriptors to yield different values than their qualitative and unweighted counterparts. This study demonstrated that weighting web properties may allow for an increased understanding of food web interactions.

Since Bersier et al.'s descriptors were not developed to detect natural regularities in community structure they may be able to detect how environmental changes can affect community structure and interactions. Albrecht et al. (2007) and Tylianakis (2007) used Bersier et al.'s weighted indices to determine the effect of land cover on two different insect food webs, and found significant differences in metric scores from webs with varying degrees of disturbed landscapes.

The goal of this study was to further understand how land use affects aquatic food webs, and to evaluate the usefulness of weighted quantitative food web metrics in describing changes in food webs accompanying anthropogenic disturbance. I focused on the relationship between fish and macroinvertebrates in second-order streams in southeastern Michigan. Most food web work done in the Great Lakes region has focused on implications of invasive species. Small warm-water streams in the region have been largely ignored despite being some of the most historically degraded aquatic systems (Cushing and Allan 2001). Food webs were described using five weighted quantitative metrics: linkage density (mean number of links for any given taxa in the web), connectance (linkage density normalized for species richness), interaction evenness (uniformity of predation), generality (mean number of effective prey taxa per predator taxa), and vulnerability (mean number of effective predator taxa per prey taxa). I expected to find differences in the metric scores between streams in developed, agricultural, and undeveloped sub-basins because of habitat loss and changes in fish and macroinvertebrate assemblages and interactions in developed and agricultural streams.

Methods

Study Sites

Study sites were located in either the Huron River or River Raisin watersheds in southeastern Michigan (figures 1 and 2). The River Raisin's headwaters are part of the Eastern Cornbelt Ecoregion located in the rolling terrain of Irish Hills in northern Lenawee County (Erickson 1995, Cifaldi et al. 2004). Geology of this

region is comprised of outwash and moraines and the river flows through a series of wetlands and lake chains (Knutilla and Allen 1975, Dodge, 1998). Meandering in a southeasterly direction the river enters the Huron-Erie Lakeplain Ecoregion where the watershed is characterized by flat terrain and intensive soybean and corn farming (Kleiman and Erickson 1996, Cifaldi et al. 2004). Geology in this area shifts to less permeable sand and clay lake plains (Knutilla and Allen 1975). The River Raisin empties into Lake Erie's western basin near the town of Monroe, MI after traveling 218 km at an average gradient of 0.605m/km (Dodge 1998). During its journey the river gathers volume from 10 major tributaries within its 2776 sq. km watershed (Kleiman et al. 1996, Dodge 1998).

The Huron River begins at Big Lake and the Huron Swamp in northern Oakland County nearly 100 kilometers northeast of the River Raisin's origin (Hay-Chmielewski et al. 1995). This region is part of the Southern Michigan/Northern Indiana Drift Plain Ecoregion; much like the River Raisin the Huron Rivers headwaters originate in an area of mixed outwash and moraines. The river courses through wetlands and lake remnants from the last glacial retreat (Michigan DNR 1977, Hay-Chmielewski et al. 1995). Downstream of the town of Dexter, the geology of the watershed becomes more impervious and transitions to till plains and then finally to sand and clay lake plains once it enters the Huron-Erie Lakeplain Ecoregion. The Huron River covers 219 kilometers at an average gradient of .557m/km before entering Lake Erie near Rockwood, less than 25 kilometers northeast of the River Raisin's mouth (Michigan DNR 1977, Hay-Chmielewski et al. 1995). The Huron River has 24 major tributaries and encompasses an area similar to that of the River Raisin, draining 2331 sq. km (Hay-Chmielewski et al. 1995). Another commonality of both watersheds is that they are highly impounded. The River Raisin watershed contains at least 50 dams (Kleiman and Erickson 1996), while the Huron River watershed has 96 dams (Hay-Chmielewski et al. 1995).

Despite the River Raisin and Huron River watersheds sharing similar geographical and geological characteristics land use within the watersheds varies substantially (figure 3). As of 2000, the Huron River

watershed was home to approximately 406,000 people while the River Raisin watersheds population is approximately one-third this amount (Cifaldi et al. 2004). The Huron River watershed is located on the fringe of the continually expanding development of suburban Detroit, and also contains small cities such as Ann Arbor and Ypsilanti (Cifaldi et al. 2004). Twenty-eight percent of the land in the basin is developed. In contrast, the River Raisin watershed does not contain any large urban or suburban population centers and only 12% of the land is developed (Infante, unpublished manuscript). Despite having substantially more urban and suburban development, the Huron River watershed also has almost twice the percentage (40% vs. 22%) of forest or wetland land coverage. Agriculture dominates land use within the River Raisin watershed accounting for nearly 63% of the total. Agriculture is also prevalent in the Huron River Basin, accounting for 25% of land use (Infante, unpublished manuscript).

Using 1995 SEMCOG land cover data for Huron River sub-basins and 1978 MIRIS land cover data for the River Raisin sub-basins (Cifaldi et al. 2004) I classified potential study sites as agricultural, developed, or undeveloped land cover composition (table 1). Aerial photographs taken in 2001 helped verify the accuracy of the land use data (Infante unpublished manuscript). Three streams were chosen to investigate for each landscape category.

All streams are second order tributaries of the Huron River or River Raisin. Agricultural sites are located in the River Raisin watershed with sub-basin agricultural land cover ranging from 31% to 74% (mean=59%). Geology of these sub-basins is dominated by fine-end moraines and outwash. All developed sites are located in the Huron River watershed. Commercial and residential development of these sites ranges 34% to 40%. The undeveloped study sites are also located in the Huron river watershed, and undeveloped land cover of the sub-basins ranges from 66% to 76%. Coarse end moraines and outwash are the primary geologies of both developed and undeveloped sites.

To score habitat quality at each site and compare local habitat conditions between stream categories I used the EPA's Rapid Habitat Assessment for low gradient streams.

Invertebrate Sampling and Analysis

Invertebrates were sampled at all sites during May 2007. Invertebrate sampling was repeated at one developed (site 164) and one undeveloped site (site 161) in September 2007 because several corresponding fish samples from these sites were lost due to contamination. No distinct differences were observed in invertebrate communities between the two time periods. At each site I sampled a 100 m reach of the stream using D frame nets and conducted a visual inspection of the organic and inorganic habitat present. Approximately 150-300 macroinvertebrates from each study site were preserved in ethanol (95%) for identification to the family or lowest practical taxonomic level using a dissecting microscope. Invertebrate communities were used to calculate a family-level biotic index (FBI) score, which indicates the extent of organic pollution at each site (Hilsenhoff 1988).

Fish Sampling and Gut Content Analysis

I sampled fish communities at all nine sites in June 2007. Sampling was repeated at developed site 164 and undeveloped site 161 in September 2007 because of the previously mentioned sample contamination at these sites. Since no temporal changes were observed in the macroinvertebrate communities for these sites I assumed there would be no significant changes in diet due to sampling date. Sampling was conducted using a backpack electroshocker (Smith-Root Inc. LR-24 Electrofisher) for a single pass effort in the same 100m reach sampled for invertebrates. All captured fish were immediately placed in a recovery tank, until identified and sorted by species, weighed to the nearest 0.1 gram, and measured (total length) to the nearest 1.0mm. Fish not used for gut content analysis were immediately released. A minimum of seven, and a maximum of fifteen, fish

from each of the five most prevalent species at each site were euthanized with MS-222 following guidelines provided by the University Committee on Use and Care of Animals, then preserved in ethanol (95%). Body cavities of fish with lengths > 100 mm were cut open to facilitate preservation of stomach contents (Winemiller 1990, Feyrer et al. 2003,). Stomachs from preserved specimens were removed and contents were examined with a dissecting microscope. Diet items for each fish were identified to family or lowest practical taxonomic level, sorted, and counted following widely used methods (Winemiller 1990, Hartman et al. 1992, Feyrer et al. 2003).

Length measurements were taken for intact diet items to determine dry biomass according to length: weight regressions defined by Benke (1999) and Nalepa and Quigley (1980). Dry mass was based on the average length of a taxon found in the diets, calculated from a subset of the taxa. Average length of each macroinvertebrate family collected during field sampling was calculated and used to supplement diet data when diet items were not found intact.

Food Web Construction and Analysis

Food webs for each site were constructed based on fish species examined for gut content analysis and invertebrates found in the diets. Webs contain three trophic levels; non-predatory invertebrates, predatory invertebrates, and fish. Quantitative links between fish and macroinvertebrates were created by calculating the proportion, based on biomass, which a prey species contributes to the total diet of fish in the community. The size of the arrows in the food webs is scaled to this proportion. Since a diet analysis of predacious invertebrates was not conducted during this study only energy flowing from invertebrates to fish is represented, despite displaying three separate trophic levels.

To analyze food webs for each site I used a series of weighted quantitative metrics described in Bersier et al. (2002), Tylianakis (2007), and Albrecht et al. (2007). These food web descriptors have an advantage over

qualitative metrics commonly used in food web studies because they account for the uneven flow of biomass in trophic links (Bersier et al. 2002). Sensitivity analysis by Bersier et al. (2002) showed quantitative metrics required less sampling effort, and were more robust (Bersier et al. 2002). In order to calculate these metrics I first developed predation matrices for each site following the procedure of Bersier et al. (2002). In the matrices, columns represent predator *P* and rows represent prey *N*. Cells indicate the biomass of prey *N* consumed by predator *P* at site *X* (table 2). From these matrices I computed five quantitative descriptors: linkage density (LD), connectance (C), interaction evenness (IE), generality (G), and vulnerability (V).

Once these metrics were calculated for each site I used SPSS statistical software package to perform the Kruskal-Wallis test to identify significant differences in biotic community properties and metric scores between land use groupings. This test was also used to detect differences in the biotic communities and metric scores based on EPA habitat scores. Sites were broken into two groups; EPA score ≥ 150 and score ≤ 149 . Analysis at this scale accounts for local habitat variation which may not be detectable when grouping streams by sub-basin land use. Nonparametric one way analysis of variance was used because the small sample size did not allow the use of an ANOVA.

Results

Rapid Habitat Assessment

Local habitat scores varied widely across the nine sites, with the lowest score (88) coming from a highly channelized agricultural stream (site 207) and the highest score of 195 from an undeveloped section of the Huron River's headwaters (site 152) (figure 4, table 3). Habitat scores also varied widely within replicates of a given stream category. The agricultural sites had the highest variance, ranging from a low of 88 to a high of 181 (figure 4, table 3). Developed sites scores ranged from 122 to 186, and undeveloped sites had the lowest variance with scores ranging from 175 to 195 (figure 4, table 3).

The agricultural streams all lost a substantial number of points for having less than optimal channel sinuosity (table 3). Channel sinuosity was rated poor at two of the sites and suboptimal at the third. However, one agricultural stream site received optimal habitat condition scores for eight of the ten categories (site 214). The other two agricultural sites received lower scores and were categorized as having poor to suboptimal conditions for most habitat parameters. Along with channel sinuosity, pool variability, bank stability, and channel alteration also scored poorly for agricultural streams.

Similar to the agricultural streams, streams in developed settings received low scores in the channel alteration parameter (table 3). Poor channel sinuosity was not as problematic at the developed sites, but poor riparian vegetative zone widths were more pronounced. Unlike agricultural streams, however, there was little evidence of erosion, and all sites had optimal scores in the bank stability parameter.

All undeveloped sites received high scores for local habitat and showed only minimum signs of degradation (table 3). Two of the sites were characterized as having suboptimal pool substrate because they lacked gravel. These sites also had suboptimal riparian vegetative width. The lowest scoring undeveloped site also scored in the suboptimal range for the epifaunal substrate/available cover, and pool variability parameters. All other parameters at these three sites had optimal ratings.

Habitat degradation varied widely among the three stream classifications, however, channel flow status had optimal ratings at every site, and no deductions were assessed for this parameter (table 3). Sites were visited throughout the year and appeared to always have sufficient flow.

Biotic Community

A total of 2632 invertebrates were identified from nine sites during the community survey (table 4). By count, Hydropsychidae was the dominant family, and accounted for 15.3% of invertebrates identified. Other

dominant taxa, by count, included Chironomidae (14.2%), Simuliidae (10%), Isopoda (6.1%), and Philopotamidae (6.0%).

After pooling data within replicates of stream categories, 45 different invertebrate taxa were identified at undeveloped stream sites (table 4), with Hydropsychidae (15.1%), Philopotamidae (13.9%), Chironomidae (7.3%), and Simuliidae (7.3%) being the dominant taxa (table 4). Thirty-six different invertebrate taxa were represented in the agricultural stream surveys with Chironomidae (25.6%), Hydropsychidae (21.5%), and Simuliidae (12.7%) the most common taxa by count. Developed sites had the poorest macroinvertebrate richness with only 35 taxa. Isopoda (16.9%), Simuliidae (9.4%), and Hydropsychidae (9.2%) were the most abundant invertebrate groups at these sites. No significant differences in invertebrate taxa richness were found using the Kruskal-Wallis test based on land use category ($\alpha=0.10$, $p=.193$). However, when the same test was run grouping streams by EPA habitat scores (score ≥ 150 , ≤ 149) invertebrate taxa richness was significantly higher ($\alpha=.10$, $p=.020$) at sites with EPA habitat scores ≥ 150 .

The water quality sensitive taxa Ephemeroptera, Plecoptera, and Tricoptera (EPT), were most prevalent at the undeveloped sites (table 4). Nineteen different EPT families were found at the three undeveloped sites and these accounted for 62.6%, 61.6%, and 32.2% of invertebrates at each site respectively. Twelve EPT families were identified at the developed sites, however, no EPT taxa were found at developed site 159. Isopoda and Turbellaria were the dominant invertebrates (49.7% and 25.3% respectively) collected at this site. At the two developed sites (sites 164 and 118) where EPT taxa occurred, they accounted for 53.3% and 45.4% of invertebrates respectively. Of the plecopterans collected during this study, 69.1% were found at these two sites.

Diversity and quantity of EPT taxa was substantially less at agricultural sites (table 4). Only nine different families were represented among the three sites. These EPT families contributed to 57.6, 18.8, and 3.4 percent, respectively, of agricultural site invertebrate communities. Despite having a high percentage of EPT

taxa at agricultural site 262 it was dominated by one family, with Hydropsychidae accounting for 75.6% of the EPT families identified. Dipterans were the dominant order at agricultural sites and nine different families were collected. The majority of taxa were from the families Chironomidae and Simuliidae.

From the invertebrate data, FBI's were computed for each site (figure 5). FBI scores varied widely between and within land use categories. Both the lowest (3.4) and highest (7.1) scores came from developed sites. The undeveloped sites had the lowest variation with scores ranging from 3.8 to 4.8. Agricultural sites had slightly higher scores ranging between 4.6 and 5.7. There was no significant difference ($\alpha=0.10$, $p=0.393$) in FBI scores between agricultural, developed, and undeveloped sites. Grouping sites by EPA habitat score (score ≥ 150 , ≤ 149) also did not show a significant difference ($\alpha=0.10$, $p=0.121$) but, did indicate a stronger pattern than grouping sites by sub-basin land use.

During electroshocking surveys, 902 fish were collected (table 5). Agricultural streams had the highest numbers of fish collected per 100 m effort with 204, 155, and 68 captured at the three sites respectively. Despite having high numbers of fish at agricultural sites, diversity was lowest, with only 13 species represented. Undeveloped sites had the highest fish diversity (21 species), followed by developed sites (19 species).

Creek chubs (*Semotilus atromaculatus*) were common across all three stream categories and 184 individuals were captured at eight sites (table 5). White suckers (*Catostomus commersonii*) were found at seven of the nine sites but not in high numbers. Creek chubs and western blacknose dace (*Rhinichthys obtusus*) were dominant in the agricultural streams with 139, and 67 individuals collected respectively. Green sunfish (*Lepomis cyanellus*), creek chubs, and *Luxilus* spp. were the dominant taxa at developed sites, while bluegills (*Lepomis macrochirus*), and fantail darters (*Etheostoma flabellare*) had the highest numbers at undeveloped sites. Similar to the invertebrate community analysis there was no significant difference in fish species richness when sites were grouped by land-use category ($\alpha=0.10$, $p=0.285$). However, also similar to the invertebrate

communities, fish species richness was significantly higher ($\alpha=.10$, $p=.091$) at sites with EPA habitat scores ≥ 150 .

Diets and Web Metrics

Food webs were constructed for each individual site using the 410 diets examined (figures 6-14). The diets showed decapods to be the dominant food source across stream categories. They were found in diets at all but one of the nine sites, and represented 52% to 98% (mean=74%) of biomass consumed at these locations. There was no significant correlation between stream category and contribution of decapods to diets. The site where no decapods were identified in the diets was in a developed setting (figure 13). Although decapods accounted for the majority of biomass consumed, consumption of this prey was not distributed evenly within fish communities. Despite being eaten by a variety of species across sites, they were only consumed by one or two species within a given site.

Terrestrial insects, Hymenoptera, were also a primary food source across all stream groupings, and were found in diets at eight of the nine sites (figures 6-14). Within these sites the contribution of Hymenoptera to fish diets ranged from 2% to 44% (mean 16%) of biomass consumed. Unlike decapods, terrestrials had a more even distribution of predation, and were preyed upon by multiple species in each food web.

Chironomids were also commonly found in diets and were consumed by every predator species examined except for rock bass (*Ambloplites rupestris*) at site 161 and *Luxilus sp.* at site 118 (figures 6-14). Despite high rates of consumption, they only contributed a small proportion of the biomass consumed in each web; less than 5% at 5 sites, between 5%- 10% at 2 sites, and between 10%-15% at 1 site. At the site where no decapods were identified in the diets, chironomids accounted for approximately 30% of biomass consumed (figure 13). Similar to Decapod consumption biomass of chironomids was not significantly different between stream groupings.

Based on visual inspection, food webs varied dramatically both within and between stream categories. However, the Kruskal-Wallis test found no significant relationship ($\alpha=0.10$) between the weighted quantitative metrics and stream category (table 6). Connectivity was the only metric that indicated a possible trend between the metrics and stream category ($\alpha=.10$, $p=.148$). All other metrics were highly insignificant, with test scores ranging from $p=0.329$ to $p=0.491$. Grouping by EPA habitat scores also did not show any significant differences.

Discussion

Invertebrates

Some commonalities were observed in the invertebrate communities. The families Hydropsychidae, Chironomidae, and Simuliidae were abundant across all land use categories, but with varying rank (table 4). Specifically, Simuliidae was dominant at developed sites, Chironomidae at agricultural sites, and Hydropsychidae at undeveloped sites. These are fairly common taxa in general and slightly to very tolerant of degraded water quality and habitat conditions, thus it is not surprising they were abundant at all streams regardless of land use. However, it is likely that chironomids are actually the most abundant invertebrate across the stream categories, but, because of their small size they are often missed or destroyed during common sampling techniques, leading to lower estimates of abundance.

Although several families were common across the stream categories they were not always found in equal proportions within replicates of stream categories. Hydropsychidae accounted for a substantial proportion of invertebrates collected in agricultural drainages, however (80%) of this family was found in a localized section of site 262 (table 4). The section was filled with rip-rap, creating structure for macroinvertebrates in a sand bottom section of stream where there was otherwise little substrate for most macroinvertebrates. The high prevalence of this moderately intolerant family produced an FBI score lower than expected based on this sites

local habitat assessment score. It is plausible to think that sampling upstream or downstream of the current section may have resulted in a poorer FBI score.

The developed stream category also was dominated by a single taxon that was present primarily at one of the sites (table 4). Isopoda represented 17% of invertebrates collected in developed streams, however 151 of the 161 Isopoda collected were found at site 159. Isopoda are a tolerant taxon and the FBI score from this site indicated poor water quality and substantial organic pollution (figure 5). We expected tolerant taxa such as Isopoda to be more common throughout the developed sites, and correspondingly higher FBI values.

Interestingly FBI scores at the other two sites (sites 118 and 164) indicated good to very good water quality with minimal organic pollution. One of these sites (site 118) had the best FBI score for all sites in the study and, the very intolerant taxa Perlidae was common here. Despite being categorized as developed and the prominence of residential land use both these streams also scored well in rapid habitat assessments. Similar to results of Infante's (unpublished manuscript) finding this suggests that macroinvertebrates respond more to local habitat conditions than to larger scale ones. Because of the limited mobility of most invertebrates they are very susceptible to local disturbances, therefore categorizing streams at a smaller scale may lead to more insightful information when examining aquatic invertebrate communities. Another plausible explanation for the prevalence of intolerant taxa at developed sites is that the amount of development at the present study sites may not have reached a threshold that caused significant degradation to the stream. Despite having substantial development in the sub-basins of these streams natural land cover was also common (table 1).

Fish

Although there were no significant differences in fish species richness between stream categories the developed and undeveloped sites had higher fish diversity than agricultural sites. This difference in diversity may be the result of habitat loss. Overall, the agricultural streams scored poorly in the EPA habitat assessment

(figure 5), which was found to correspond with decreased diversity in this study. As mentioned previously, agricultural streams were straightened to varying degrees, and as a result lacked different depth regimes and in-stream structure necessary for promoting diversity. The agricultural streams also lacked bank side trees, which are an important for contributing coarse woody debris, which provides cover for many fish species.

Connectance to lentic water may have also influenced fish communities. Common lacustrine species such as bluegills (*Lepomis macrochirus*), green sunfish (*Lepomis cyanellus*), and warmouth (*Lepomis gulosus*) were more prevalent in the developed and undeveloped sites compared to the agricultural streams (table 5). While all undeveloped and developed sites had relatively close connections to lentic systems, agricultural sites had no relatively close connections. Unlike the agricultural streams many of the undeveloped and developed streams also had slow deep pools preferred by lake dwelling species. In another study Infante (unpublished manuscript) found centrarchides most common in southeastern Michigan streams with coarser surficial geology, which is characteristic of the undeveloped and developed sites in this study.

Although the lowest mean fish species richness was found at the agricultural sites the lowest diversity for an individual site was at developed site 159 (table 5). Only five different species were collected at this site and tolerant green sunfish and bluegills accounted for 90% of the catch. This site also had the lowest invertebrate diversity (table 4) and highest FBI score indicating very substantial organic pollution (figure 5). Unlike the other developed sites the local habitat was substantially degraded (table 3). Rip-rap and train tracks paralleled one bank. The other bank had little to no riparian buffer and a manicured lawn bordered the stream. These forms of local land cover manipulation are known to be destructive to fish and likely are the causes of poor diversity of this site.

The described differences within replicates of developed sites occurred despite all sites having similar sub-basin land use (table 1). However, some development has only minor ecological impacts or is done in a way that minimizes these impacts while other development can have substantial negative effects on aquatic

ecosystems. The developed sites in this study illustrate this concept. Development at two of the sites (sites 118 and 164) is characterized by residential housing in natural settings with few anthropogenic impervious surfaces, while one site (site 159) has a mix of industrial and dense residential development with many impervious surfaces. These differences appeared to result in substantially different impacts to the ecological health of the streams despite a common classification of land cover applied at the sub-basin level. This result is further evidence that sub-basin land cover classification may be insufficient and looking at smaller scale descriptions of habitat is important when investigating how anthropogenic land use affects small stream communities.

Food Webs and Metrics

Based on observed differences in macroinvertebrate communities between stream categories it seemed likely that there would be corresponding differences in the dominant prey consumed by fishes. Although it was expected that certain species would be more common at all sites, an *a priori* hypothesis was that presence of additional species at undeveloped sites would dilute the importance of common warm water invertebrates such as chironomids in fish diets. Instead, Decapoda, terrestrial Hymenoptera, and Chironomidae were found to be main prey items across all three stream categories, with decapods being the primary prey source. The biomass of a single Decapod can be several orders of magnitude greater than other commonly consumed invertebrates, leading to its dominance in the food webs despite being consumed in low numbers and by only one or two species at each site.

Because different Decapod families have a broad range in tolerance to organic pollution and human disturbances they are common in a wide variety of warm water streams throughout the United States (Peake et al. 2004). Although many species of decapods are threatened, the tolerant taxa of this order persist in streams experiencing agricultural and urban development. Their abundance makes them an important prey source for fish in both degraded and healthy streams with varying land cover, as was found in this study.

Similar to decapods, chironomids are often abundant in both healthy and degraded aquatic ecosystems and form an important part of the prey base for many fish species. Chironomidae biomass consumption was much lower than Decapoda consumption but was still important, accounting for as much as 30% of fish diets at one site. Chironomids were the dominant food item by number and preyed upon by almost every fish species examined. Since most fish at these sites are gape limited in decapod consumption, and terrestrials are a seasonal food source, chironomids may be the most important prey item across all stream categories.

Similar to the studies of Allan et al. (2003) and Cloe and Garman (1996) the present study found terrestrial invertebrates (Hymenoptera) to be an important food source. In many forested streams terrestrial invertebrates may constitute approximately 50% of fish diets (Saunders and Fausch 2007). Because agricultural and urban development can reduce riparian vegetation and thus negatively affect terrestrial invertebrate inputs to aquatic food webs (Sweka & Hartman 2008) I expected to find higher terrestrial invertebrate consumption in natural streams compared to developed and agricultural sites. Riparian vegetation and pesticide use are limiting factors for terrestrial invertebrate inputs into aquatic food webs. Saunders and Fausch (2007) found terrestrial inputs were 2.3 times greater in streams that had restricted grazing compared with streams with intensive grazing on the adjacent land.

Contrary to this expectation, I found no differences in terrestrial invertebrate (Hymenoptera) consumption between developed, agricultural, and urban sites. As stated previously not all anthropogenic land use activities are equal in terms of ecological impact. Development and agriculture that preserves some riparian vegetation and forgoes the use of pesticides may produce little or no loss in these inputs of terrestrial prey items. Despite some of the sites having sub-optimal riparian vegetation it is likely that enough buffer was sustained to not substantially affect terrestrial inputs.

With the original expectations that predation patterns would vary with land use I also expected to see differences in multiple weighted quantitative food metrics. Two recently published studies found significant

differences in metrics between undeveloped and degraded ecosystems, however, these studies have conflicting findings. Tylianakis et al. (2007) compared insect food webs along an agricultural disturbance gradient and found higher interaction evenness and lower vulnerability, but no differences in generality, linkage density, and connectance. In contrast, a study of insect food webs among varying types of meadows found that generality and linkage density was higher in restored meadows compared to intensively managed ones, but interaction evenness was significantly lower in restored meadows, and found vulnerability did not differ (Albrecht et al. 2007).

Most of the fish species examined in this study were generalist invertivores. With higher prey diversity in undeveloped communities I expected to find a corresponding increase in generality and interaction evenness. Because of higher prey species richness in undeveloped sites and generalist feeding practices by these fish I anticipated that fish would become less dependent on a single food source, increasing the uniformity of predation. I also anticipated higher vulnerability at undeveloped sites because of higher predator richness equating to more predators for any given prey source. Although linkage density remains relatively constant in species poor food webs, species rich webs tend to have increased values for this metric (Pimm et. al 1991). Following the prediction that species richness would be higher in undeveloped settings, I expected linkage density to be highest at these study sites as well. However, I did not anticipate that there would be any significant variance in connectance within and across stream categories because this metric is standardized to control for species richness.

Despite trends showing a decline of invertebrate and fish species richness and loss of intolerant taxa with increasing anthropogenic land use there were no significant differences in the weighted quantitative metrics between stream categories (table 6). Variability in habitat quality, species richness, and metric scores among replicates of land use categories may have limited the ability to find significant differences in metrics between land use categories. However, inter-category variability is most likely not the reason why no

significant differences in metric scores were observed in the intra-category analysis. Even when local habitat quality conditions were controlled for by grouping according to EPA habitat assessments, metric scores between groups were highly insignificant.

A more plausible explanation for a relationship between metric scores and land use or habitat quality -is the unexpected dominant effects of decapod consumption on metric scores. Decapoda biomass consumption was substantially higher than all other prey sources and was the primary driver of the metric scores. Because of the dominance of decapods as a prey source, differences in their consumption among replicates can explain high inter-category metric variance.

Generality was lowest at sites where decapods were the only substantial prey taxa, and highest at sites where the dominant predator had multiple substantial prey sources (figures 6-14, table 6). As mentioned previously, Hymenoptera and Chironomidae were other significant prey sources and contributed to increased generality at several sites. This finding is exemplified in the undeveloped stream category comparing site 152 (figure 11) to site 104 (figure 9). At site 152, three of the fish species each had two major prey sources and multiple minor ones, while at site 104 decapods were the only major prey taxa for both fish species in the web. The feeding pattern of fish at site 152 resulted in a generality score 60% higher than at site 104 (table 6).

Variation among vulnerability scores clearly demonstrated how decapod consumption was the primary determinant of the metric scores. Sites where decapods had multiple predators typically had higher vulnerability scores compared to sites where they were fed upon by a single fish species. This difference can be seen by comparing the developed site 164 (figure 14) to the undeveloped site 161 (figure 10). Decapods at site 164 were eaten by both horneyhead chubs and creek chubs, where at site 161 decapods were only consumed by rock bass. It is also possible vulnerability scores obtained in this study may have been affected by only selecting common predators for diet analysis and web construction. Despite existing in low numbers, some rare species can have significant impacts on community interactions.

Linkage density and connectance are both factors of generality and vulnerability, and like these metrics they varied within stream categories and showed no significant difference between stream categories. These metrics also were highly influenced by decapod consumption (table 6).

Interaction evenness was also affected by high decapod biomass consumption. An example of this effect is seen by site 104's low interaction evenness score, which can be attributed to heavy decapod consumption by predominately one species. At site 104 decapods accounted for approximately 95% of biomass consumed, with 92% of the consumption by warmouth (figure 9). In comparison decapod consumption at site 164 (figure 14) was more uniform and this site had higher interaction evenness.

The overwhelming influence that decapod consumption had on the weighted quantitative metrics is also demonstrated by the lack of decapods in the diets at developed site 118 (figure 13). Since the mass of a decapod is dramatically higher than any other invertebrate consumed, decapods diluted the importance of other invertebrates in the webs, especially for fish whose gape limited the potential for decapod consumption. Without decapods in the food web, other invertebrates such as Chironomidae, Hydropsychidae, Hymenoptera, Elmidae, and Caenidae contributed large proportions to the diets. The absence of this taxa resulted in developed site 118 having the highest linkage density, connectance, interaction evenness, and generality (table 6).

Although decapods were not present in the diets from site 118, they were found during the invertebrate collection (table 4). It is likely that fish at this site consumed decapods, and decapods played an important role in the food web, but had not been consumed at a time close to the fish sampling. Adding a temporal component to this study may enhance knowledge of feeding patterns in this stream and result in more accurate metric scores. This study only accounted for what fish had consumed in the few hours preceding their capture and a high degree of temporal equilibrium had to be assumed with the present analysis. Collecting multiple samples

over the course of a year would account for seasonal changes in feeding rate and prey selection. Increased diet data would also allow us to identify infrequent but important feeding events.

Increasing the number of streams sampled would also have benefited the study by boosting statistical power. The low “n” of this study limited the options for analyzing the data, and the ability to detect the influence of land cover on stream food webs and how they functioned. With more streams in the study it may have been possible to find significant differences in fish and invertebrate communities, local habitat scores, and FBI scores between land use categories. However, it is unlikely that increasing sample size or resolution would result in statistically significant findings for the quantitative weighted food metrics, because variation of metric values within categories was high. Despite loss of invertebrate diversity through anthropogenic land use in developed and agricultural settings the primary prey taxa remained consistent.

Conclusion

Streams in undeveloped settings showed patterns of higher fish and invertebrate diversity when compared to streams in agricultural or developed settings, and fish and invertebrate diversity was significantly higher in streams with better quality local habitat. Despite these trends high decapod biomass consumption influenced the metrics, muting the effects of anthropogenic disturbance on the food webs. Smaller fish, which are gape limited in decapod consumption, may be more dependent on various families of smaller, more tolerant taxa. However, changes in consumption of smaller macroinvertebrates could not be detected by the metrics because of the influence of decapods on metric scores. This finding indicates that these metrics may not be useful in determining the effects of land cover on aquatic food webs when large tolerant taxa are the main prey sources.

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Table 1. Sub-basin land cover data showing land use in each tributary used in the study.

Site	Stream Category		
	Developed	Agricultural	Undeveloped
Developed			
164	33.78	14.64	46.76
118	34.14	10.27	47.46
159	32.84	18.82	43.61
Agricultural			
214	10.36	68.38	21.18
262	5.01	72.53	21.32
207	4.35	78.64	16.99
Undeveloped			
152	18.43	11.07	63.83
161	10.95	25.95	61.92
104	20.36	9.46	67.45

Table 2. Example predator-prey matrix. Each cell represents the proportion a diet item contributes to the total biomass consumed.

Diet Items		Fish Species			
Order	Family	bluegill	creek chub	fantail darter	central mudminnow
Isopoda		0.0012	0.0006	0.0000	0.0000
Gastropoda		0.0000	0.0000	0.0000	0.0002
Bivalvia		0.0000	0.0000	0.0000	0.0052
Nematoda		0.0000	0.0000	0.0000	0.0002
Decapoda		0.0000	0.4187	0.0000	0.2094
Odanata	Aeshnidae	0.0000	0.0187	0.0000	0.0000
Coleoptera	Dryopidae	0.0000	0.0000	0.0000	0.0007
	Elmidae	0.0000	0.0000	0.0000	0.0006
Heteroptera	Gerridae	0.0015	0.0000	0.0000	0.0000
Diptera	Chironomidae	0.0651	0.0003	0.0108	0.0054
	Simuliidae	0.0030	0.0040	0.0007	0.0003
	Ceraptopognidae	0.0000	0.0000	0.0000	0.0001
	Dixidae	0.0000	0.0000	0.0000	0.0001
	Tipulidae	0.0000	0.0026	0.0000	0.0000
Ephemeroptera	Baetidae	0.0012	0.0000	0.0022	0.0000
	Caenidae	0.0000	0.0000	0.0000	0.0001
Tricoptera	Hydropsychidae	0.0017	0.0000	0.0149	0.0000
	Glossosomatidae	0.0000	0.0000	0.0002	0.0000
	Uenoidae	0.0000	0.0000	0.0000	0.0000
	Limnephilidae	0.0000	0.0170	0.0000	0.0000
Hymenoptera		0.0824	0.0581	0.0000	0.0727

Table 3. EPA Habitat Assessment-Individual Parameter Score. For individual habitat parameter 0-5=poor, 6-10=marginal, 11-15=suboptimal, 16-20=optimal. For total score 0-50=poor, 51-100=marginal, 101-150=suboptimal, 151-200=optimal.

Habitat Parameter	Site Number and Category								
	Developed			Agricultural			Undeveloped		
	164	118	159	214	262	207	152	161	104
Epifaunal Substrate/ Available Cover	18	19	13	20	9	9	20	18	15
Pool Substrate Characterization	20	18	13	18	12	12	20	15	14
Pool Variability	20	20	13	17	11	4	19	17	15
Sediment Deposition	18	19	18	19	14	9	19	17	18
Channel Flow Status	20	20	20	20	20	18	20	20	20
Channel Alteration	19	14	13	17	14	9	19	19	18
Channel Sinuosity	19	15	5	15	5	0	18	20	19
Bank Stability	20	20	18	19	17	10	20	20	20
Vegetative Protection	18	11	7	20	18	14	20	19	20
Riparian Vegetative Zone Width	14	9	2	16	16	3	20	15	16
TOTAL SCORE	186	165	122	181	136	88	195	180	175

Table 4. Summary of invertebrate collection

Invertebrates		Site Number and Category								
		Developed			Agricultural			Undeveloped		
Family	Order	164	118	159	214	262	207	152	161	104
Isopoda		2	3	151	0	0	4	0	1	0
Amphipoda		6	0	28	2	13	52	13	1	24
Gastropoda		12	12	2	4	3	15	0	12	4
Bivalvia		0	4	0	1	3	8	1	1	0
Hirudinea		0	1	2	0	0	2	0	0	0
Oligochaeta		0	0	7	0	0	1	3	0	0
Turbellaria		0	0	77	0	0	0	0	2	0
Hydracarina		0	0	0	0	0	0	0	1	0
Decapoda		3	4	2	0	19	2	2	2	1
Odanata	Aeshnidae	4	8	5	2	0	0	5	5	7
	Gomphidae	2	4	0	0	0	0	4	0	0
	Calopterygidae	10	17	2	21	0	11	3	4	7
	Coenagrionidae	2	0	2	0	2	22	0	1	1
	Cordulegastridae	0	0	0	0	0	0	4	0	0
Coleoptera	Elmidae	16	20	1	34	36	3	14	20	3
	Haliplidae	0	0	0	1	1	0	0	0	0
	Dytiscidae	0	0	8	1	0	0	0	0	3
	Dryopidae	0	0	0	4	0	0	5	31	7
	Hydrophilidae	0	0	0	1	1	0	0	0	0
	Pesphenidae	1	11	0	0	0	0	0	1	0
	Scirtidae	2	0	0	0	0	0	0	0	0
Megaloptera	Corydalidae	1	0	0	0	0	0	3	0	1
	Sialidae	0	0	0	0	0	0	3	1	2
Heteroptera	Belastomatidae	2	0	0	0	0	0	1	0	0
	Corixidae	0	0	0	0	12	0	0	0	0
	Veliidae	0	0	0	0	1	0	0	0	0
	Gerridae	0	0	0	3	0	0	0	0	0

Invertebrates		Site Number and Category								
		Developed			Agricultural			Undeveloped		
Family	Order	164	118	159	214	262	207	152	161	104
Diptera	Chironomidae	41	21	15	131	51	57	20	18	19
	Simuliidae	77	7	3	1	11	107	10	47	0
	Tabanidae	0	0	0	3	0	2	3	0	0
	Tipulidae	0	7	0	1	0	1	5	0	0
	Stratiomyidae	1	0	0	0	0	1	0	0	0
	Dixidae	5	0	0	6	0	0	0	0	0
	Culicidae	0	0	0	0	1	0	0	0	0
	Empididae	0	0	0	0	2	0	0	0	0
	Ptychopteridae	0	0	0	0	1	0	2	0	0
	Ceratopogonidae	0	0	0	0	0	0	0	0	1
Ephemeroptera	Baetidae	28	30	0	1	45	0	7	10	9
	Heptageniidae	23	1	0	2	3	0	10	29	2
	Caenidae	4	0	0	1	0	0	2	3	2
	Isonychiidae	0	3	0	0	0	0	0	28	0
	Leptohiphidae	11	1	0	0	3	0	0	2	0
	Ephemeridae	0	0	0	0	0	0	0	0	1
	Leptophlebiidae	0	0	0	0	0	0	57	0	1
Plecoptera	Perlidae	15	51	0	7	0	0	0	1	14
	Nemouridae	0	0	0	5	0	0	0	0	1
	Perlodidae	0	1	0	0	0	0	0	0	2
Tricoptera	Hydropsychidae	81	4	0	31	161	9	78	39	0
	Philopotamidae	50	0	0	0	0	0	1	107	0
	Helicopsychidae	1	0	0	0	0	0	0	0	0
	Limnephilidae	0	6	0	1	1	1	7	0	6
	Brachycentridae	0	2	0	0	0	0	0	14	1
	Phryganeidae	0	0	0	0	0	0	0	1	0
	Polycentropodidae	0	0	0	0	0	0	3	3	0
	Psychomyiidae	0	0	0	0	0	0	1	0	0
	Uenoidae	0	0	0	0	0	0	2	0	0
	Odontoceridae	0	0	0	0	0	0	1	0	0
	Lepidostomatidae	0	0	0	2	0	0	0	0	0
Taxa Richness		25	22	14	24	20	17	29	27	23

Table 5. Summary of fish collection

Fish Species	Site Number and Category								
	Developed			Agricultural			Undeveloped		
	164	118	159	214	262	207	152	161	104
white sucker	2	1	0	0	1	1	2	1	3
northern hog sucker	14	7	0	0	0	0	0	1	0
rock bass	1	1	0	0	0	0	0	17	3
green sunfish	0	0	51	0	3	7	7	10	1
pumpkinseed	0	0	0	0	0	0	1	2	0
warmouth	0	0	0	0	0	0	0	0	7
bluegill	0	2	25	0	0	4	39	9	38
smallmouth bass	0	0	0	0	0	0	0	2	0
largemouth bass	0	0	0	0	0	0	1	0	1
mottled sculpin	4	6	0	11	38	5	0	2	0
central stoneroller	0	0	0	3	0	0	0	0	0
spotfin shiner	0	1	0	0	0	0	0	0	0
spottail shiner	0	2	0	0	0	0	0	0	0
<i>Luxilus sp.</i>	16	16	0	0	1	0	0	4	0
honeyhead chub	28	1	0	0	0	0	0	0	0
river chub	0	0	0	0	0	0	0	5	0
creek chub	13	19	0	105	17	7	14	7	2
bluntnose minnow	0	0	2	0	1	0	0	0	0
fathead minnow	0	0	0	4	1	38	0	0	0
western blacknose dace	0	0	0	35	0	32	0	0	0
orangethroat darter	0	0	0	38	6	0	0	0	0
greenside darter	0	4	0	0	0	0	1	1	0
rainbow darter	0	1	0	0	0	0	0	4	0
Iowa darter	0	0	0	0	0	0	0	0	0
fantail darter	1	0	0	0	0	0	29	2	0
johnny darter	0	0	0	8	0	11	0	0	0
yellow perch	0	0	0	0	0	0	2	0	0
logperch	0	0	0	0	0	0	0	4	0
blackside darter	0	0	0	0	0	0	0	2	0
stonecat	1	0	0	0	0	0	0	0	0
yellow bullhead	1	0	0	0	0	0	0	0	0
blackstripe topminnow	0	0	0	0	0	0	0	0	1
central mudminnow	0	0	5	0	0	50	18	0	6
grass pickeral	0	0	1	0	0	0	0	0	0
Species Richness	10	12	5	7	8	9	10	16	9

* striped and common shiners are lumped as *Luxilus sp.* because of high frequencies of hybridization in this region

** Numbers in bold indicate species used for diet analysis. Site 164 green sunfish were contaminated and not used for diet analysis.

Table 6. Summary of weighted quantitative food web metrics

Site	Linkage Density	Connectance	Interaction Evenness	Generality	Vulnerability
Developed					
164	1.817	0.096	0.215	1.542	2.092
118	2.765	0.230	0.433	3.681	1.849
159	1.951	0.130	0.233	2.477	1.425
Agricultural					
214	1.840	0.077	0.197	1.748	1.932
262	1.453	0.145	0.091	1.103	1.804
207	2.246	0.112	0.271	3.072	1.421
Undeveloped					
152	2.143	0.086	0.256	2.206	2.080
161	1.332	0.063	0.139	1.408	1.256
104	1.348	0.079	0.074	1.380	1.317
significance (p=.05)	0.393	0.148	0.393	0.329	0.491

Study Region

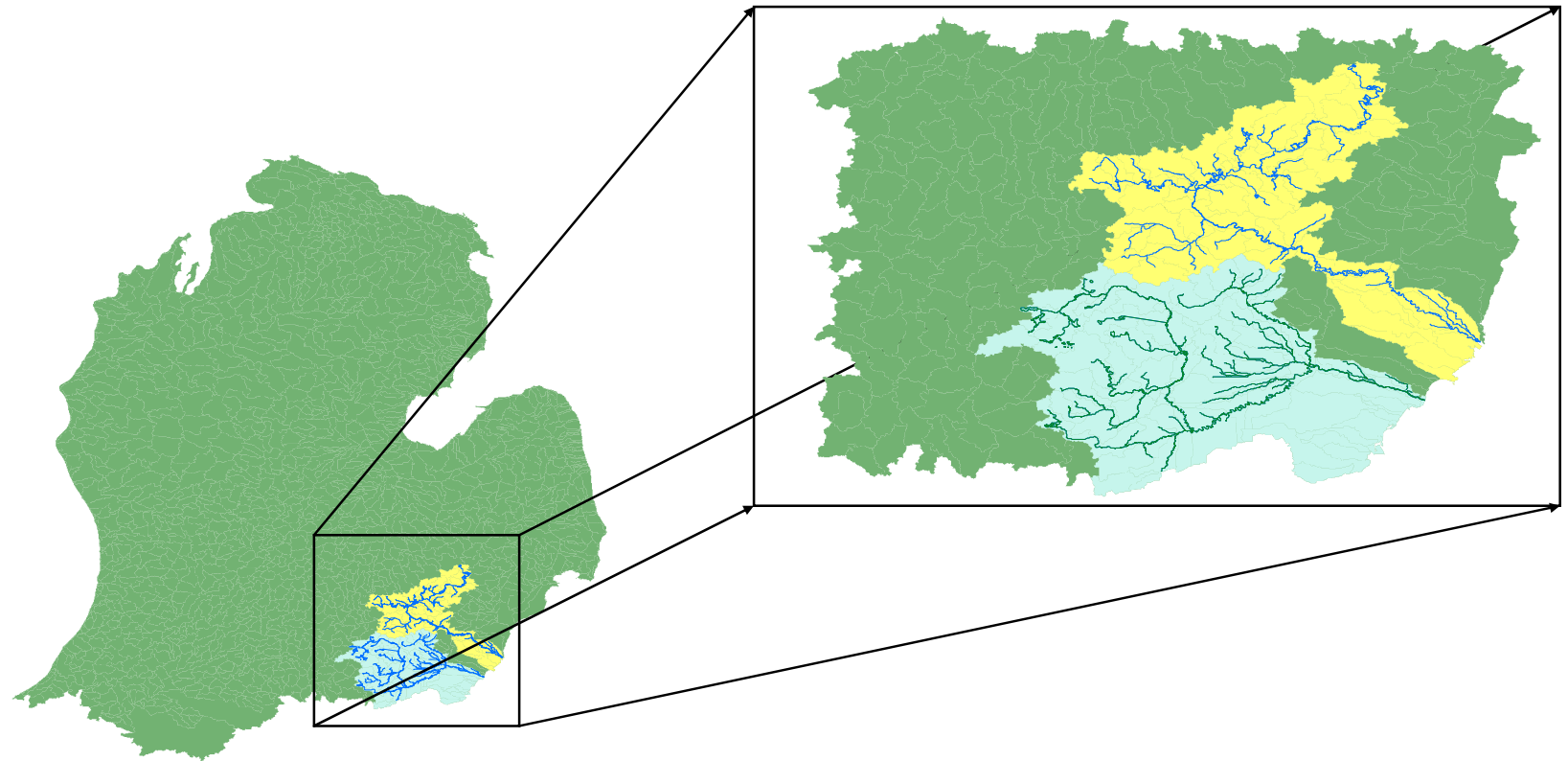


Figure 1. Study Region

Site Locations

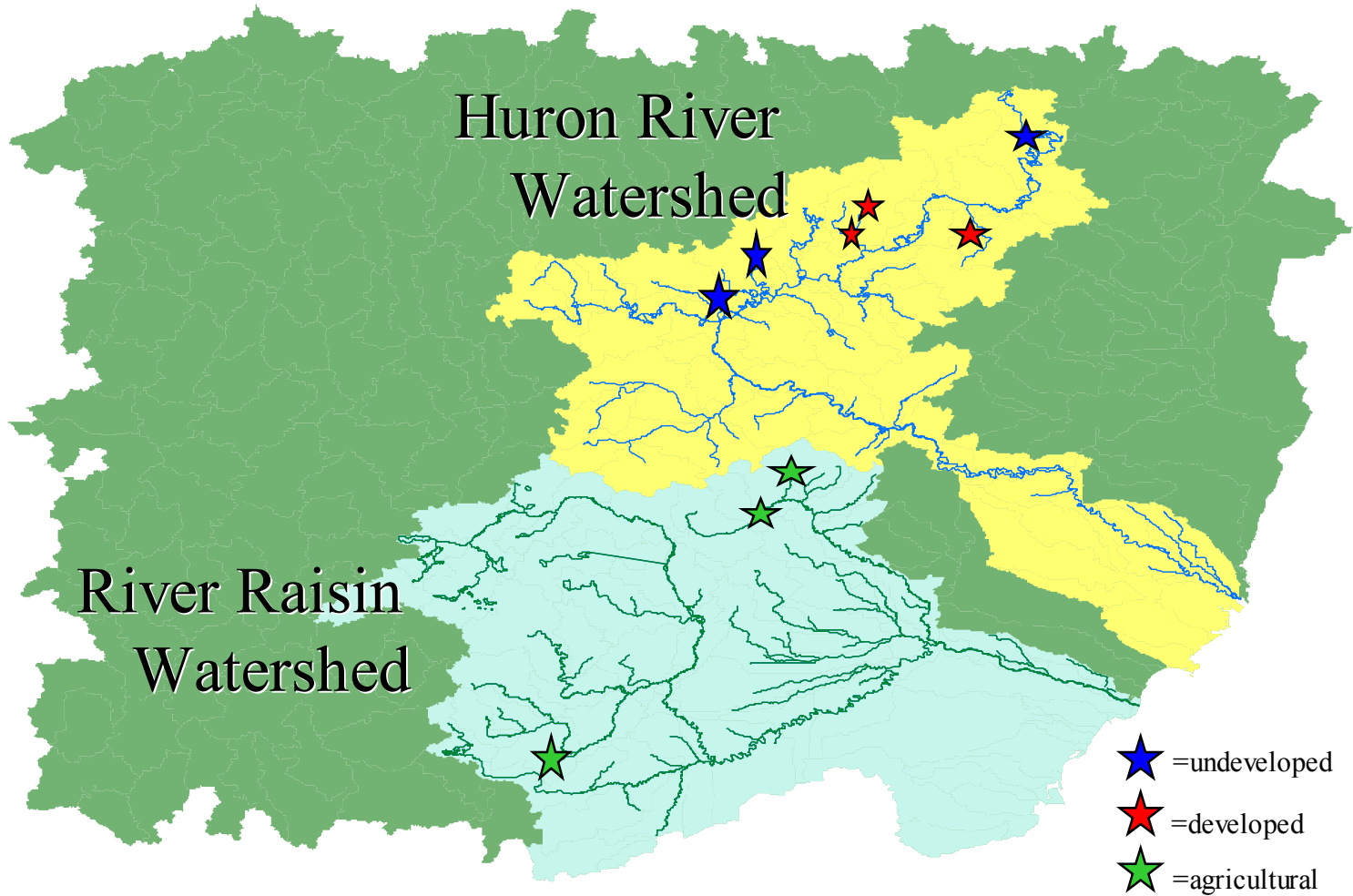


Figure 2. Site Locations

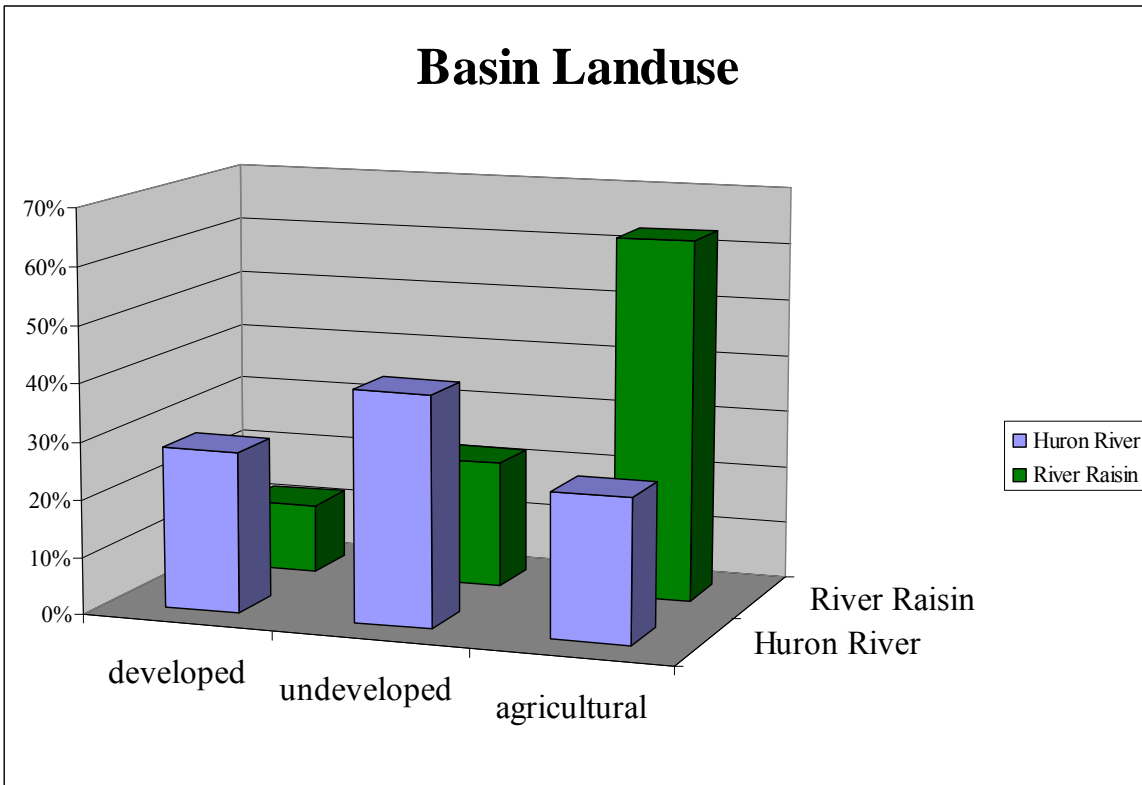


Figure 3. Land use at the basin level for the Huron and River Raisin watersheds

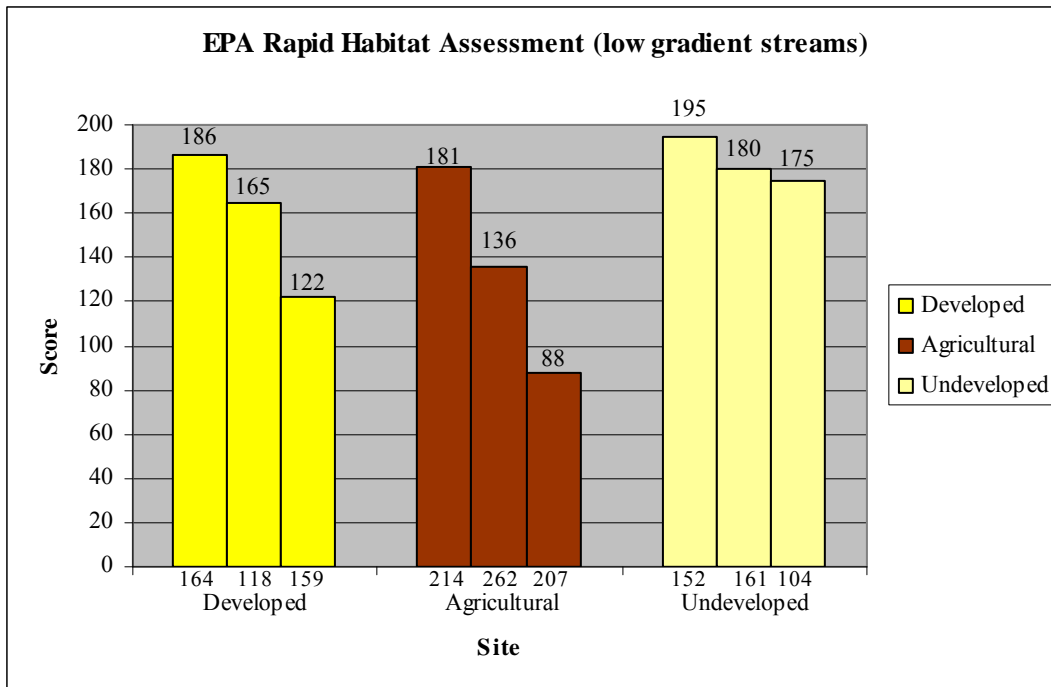


Figure 4. EPA Rapid Habitat Assessment-Total Score

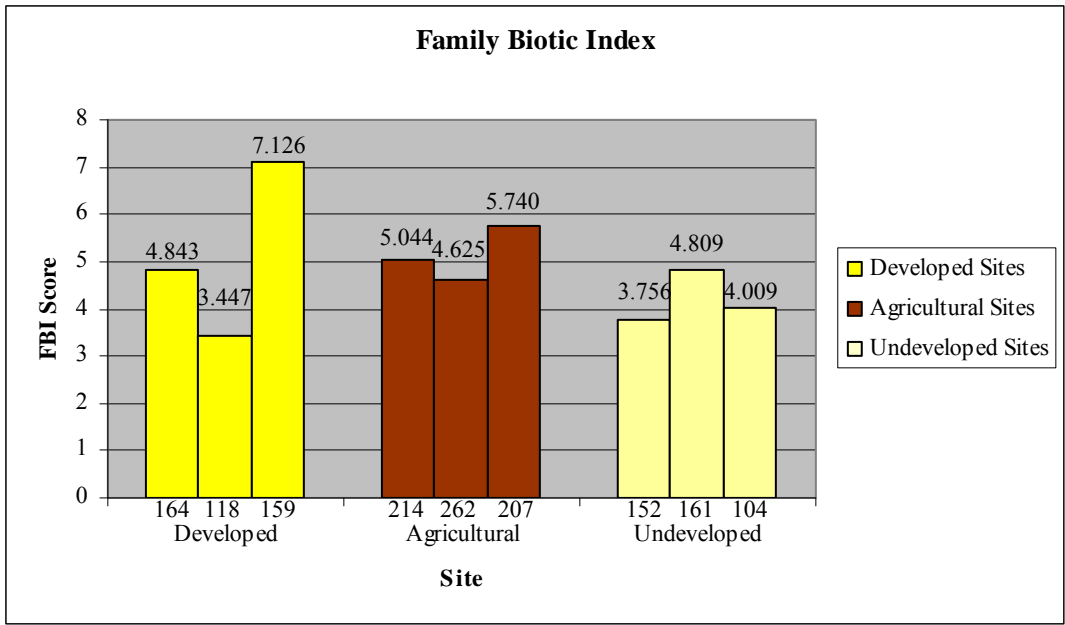


Figure 5. Hilsenhoff FBI

Figure 6. site 262. Saline River-Agricultural

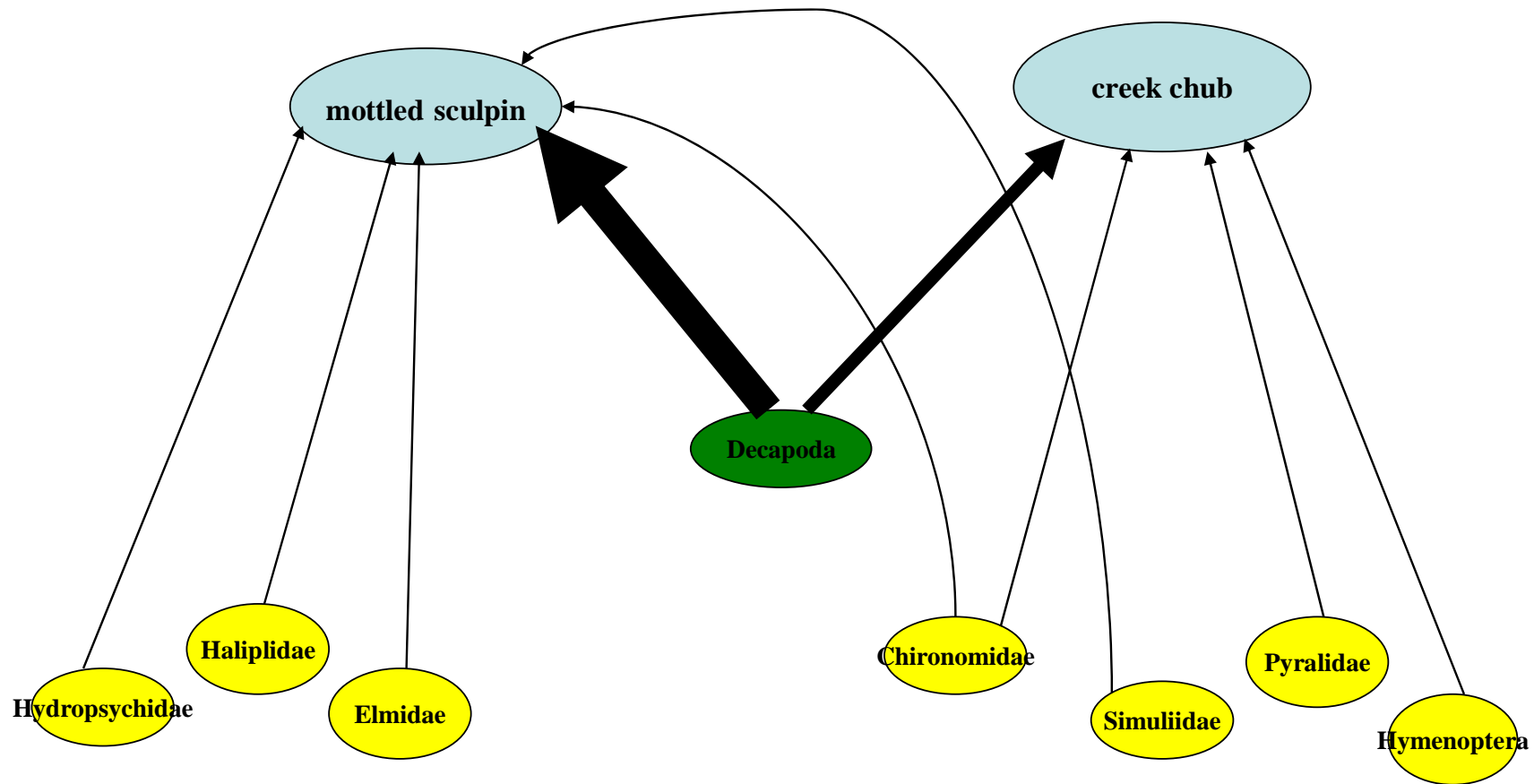


Figure 7. site 207. Stony Creek-Agricultural

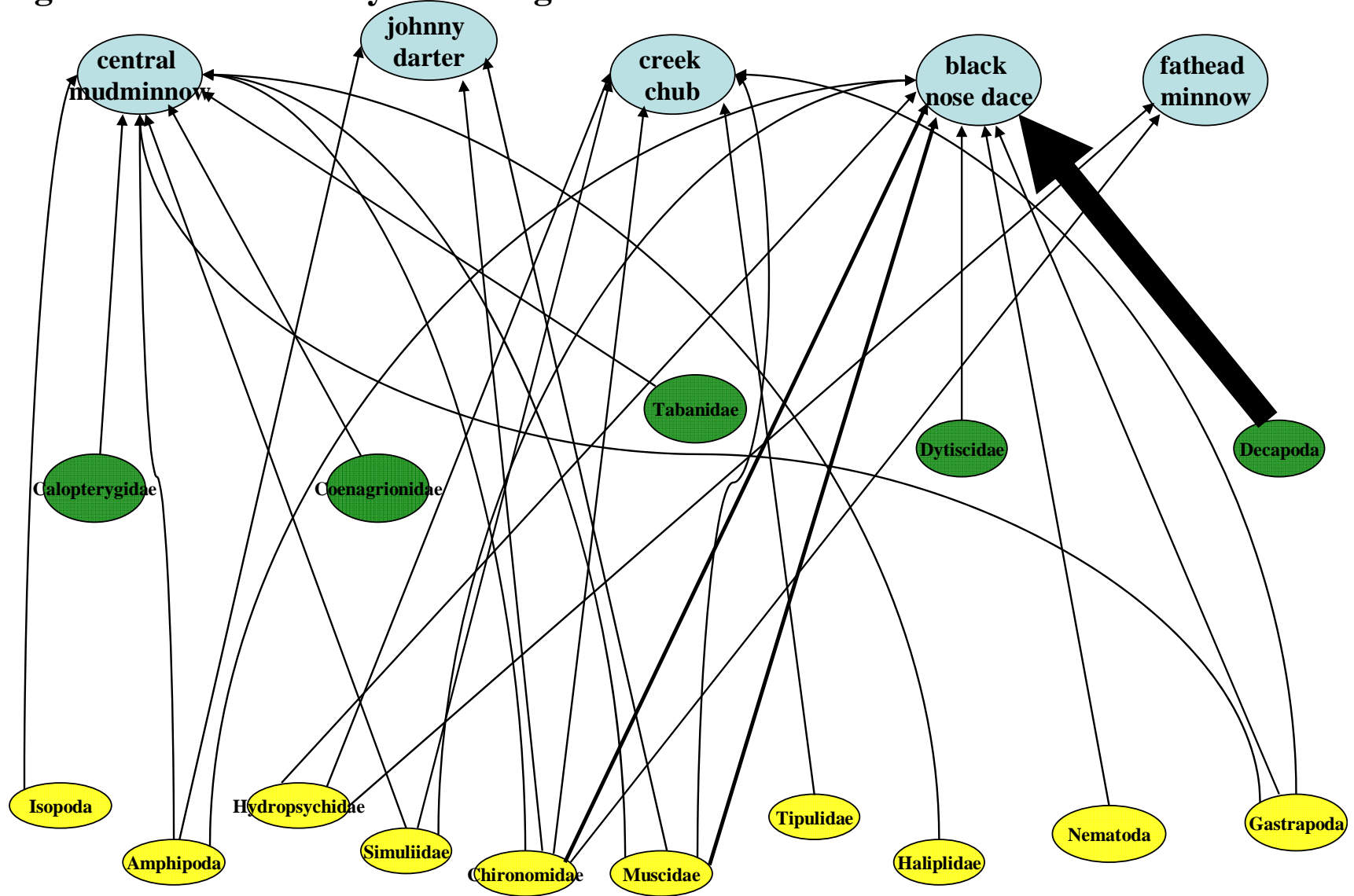


Figure 8. site 214. unnamed tributary to the Saline River-Agricultural

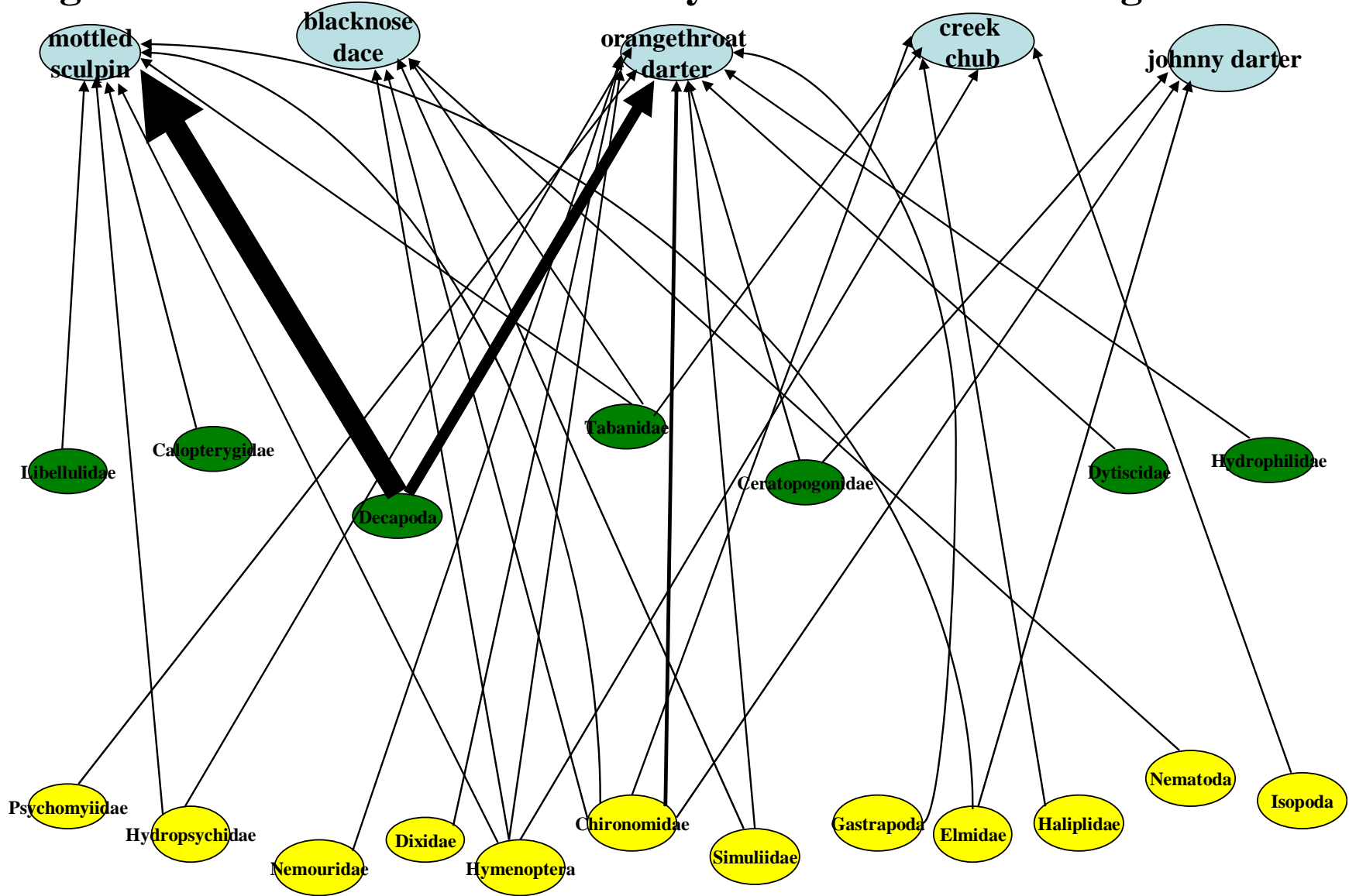


Figure 9. site 104. Hay Creek-Undeveloped

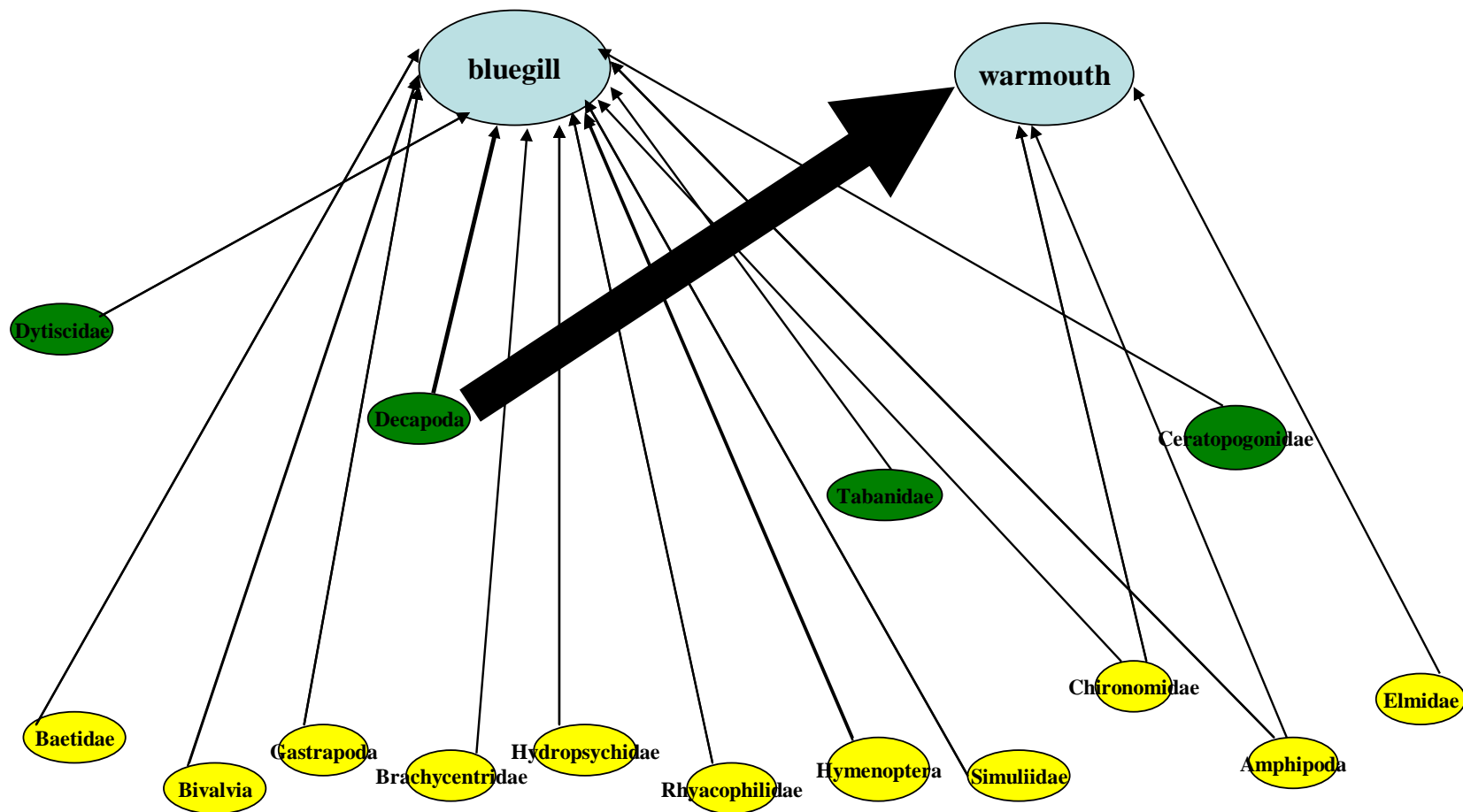


Figure 10. site 161. Honey Creek-Undeveloped

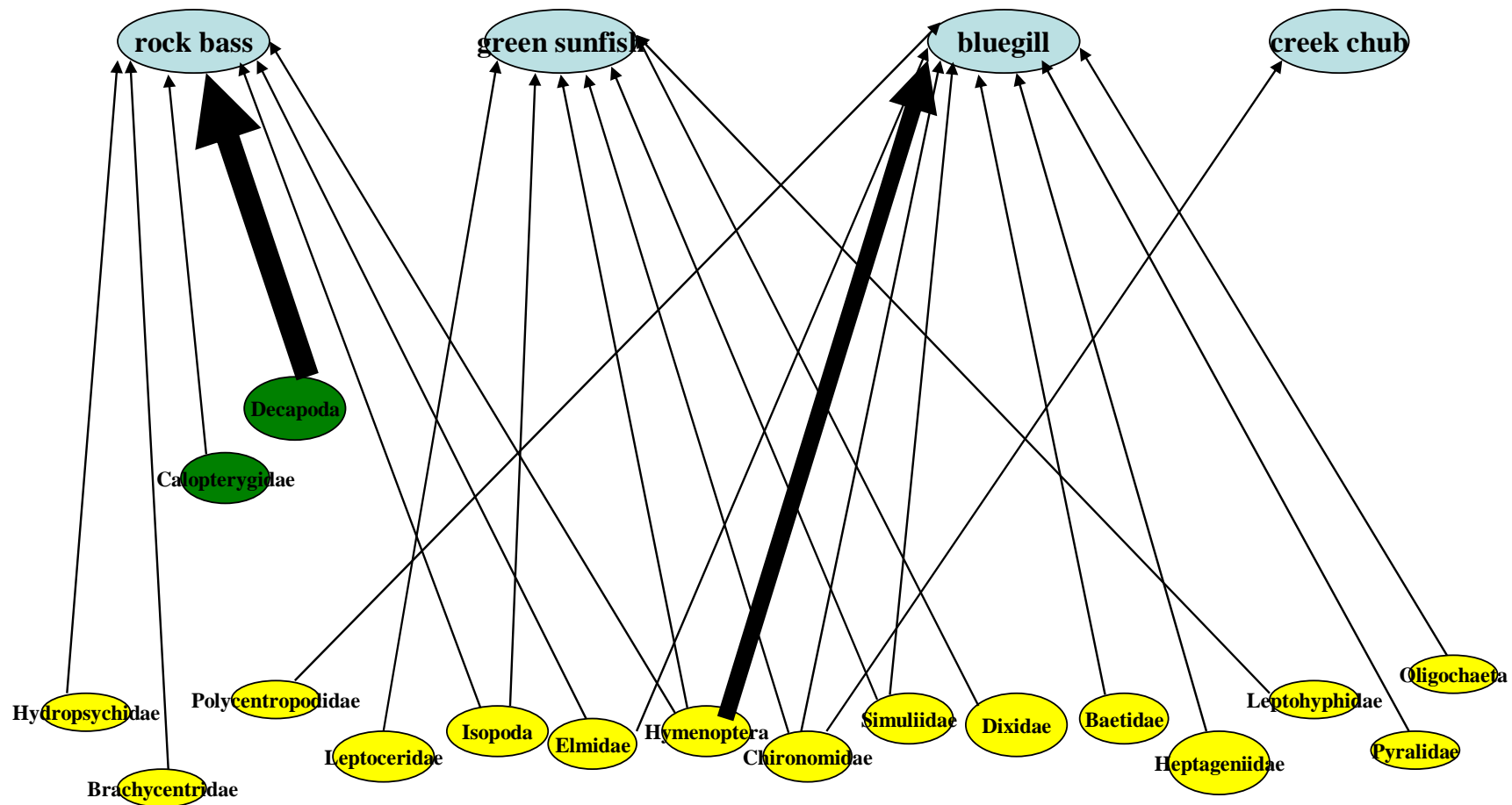


Figure 11. site 152. Huron River-Undeveloped

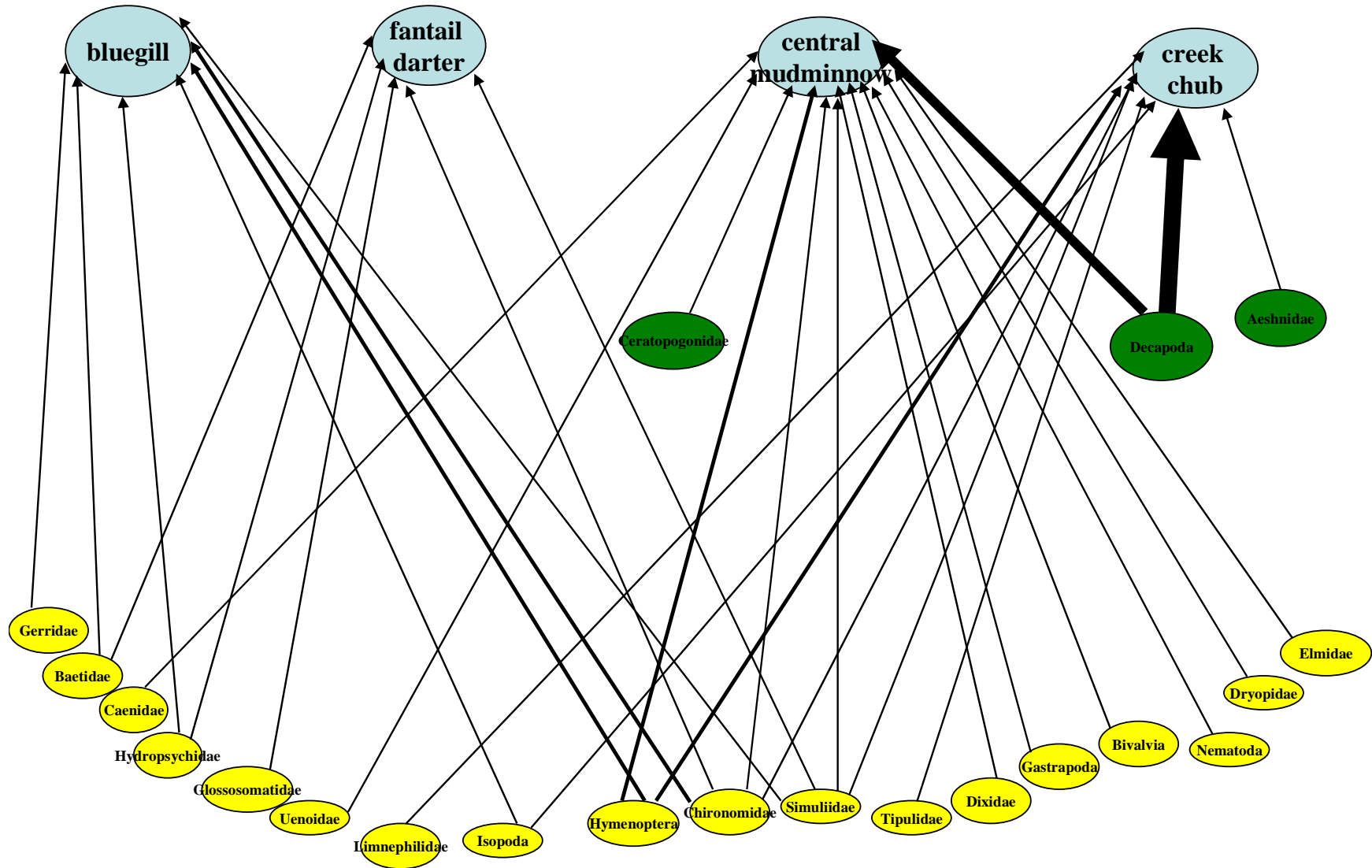


Figure 12. site 159. Norton Drain-Developed

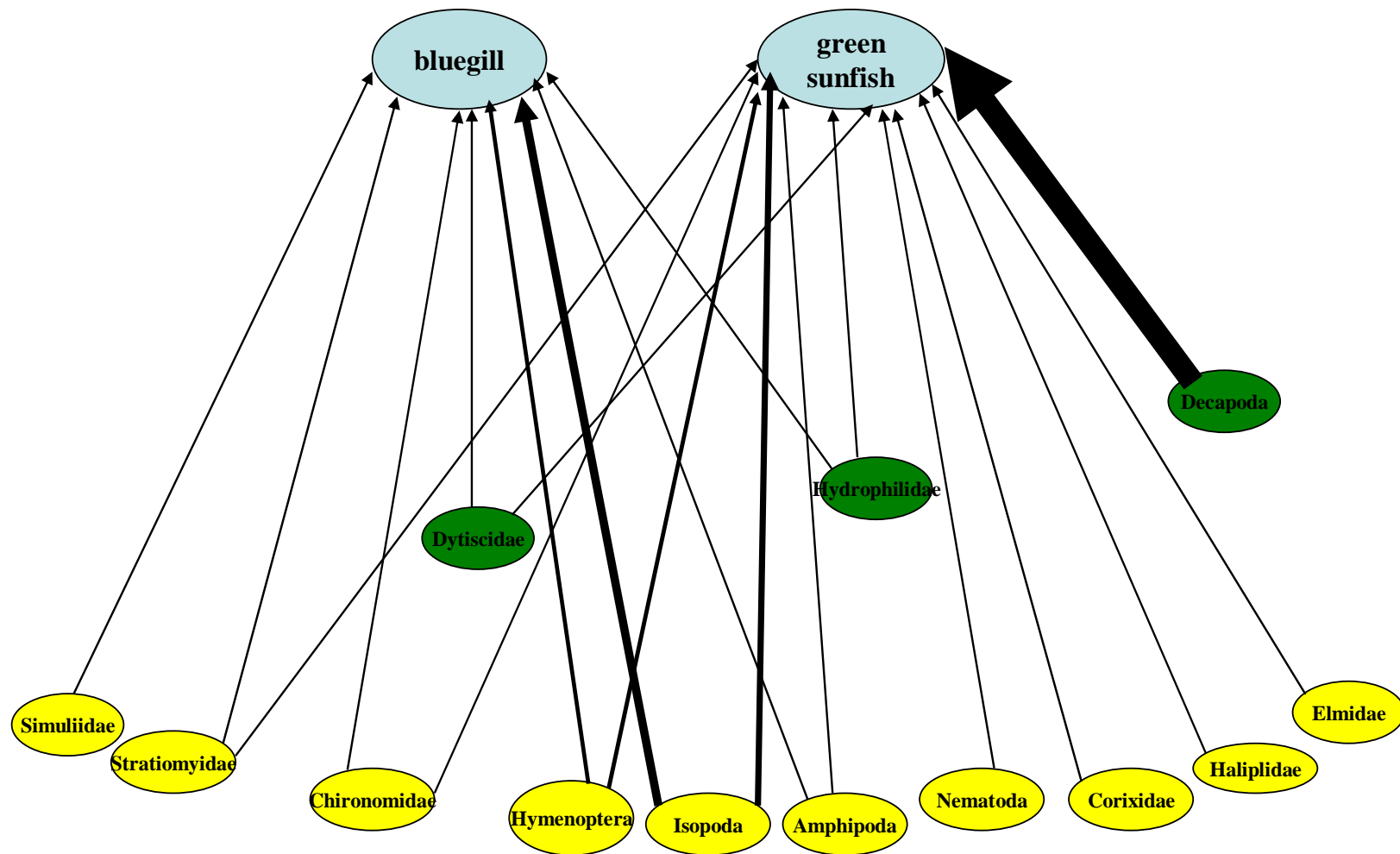


Figure 13. site 118. Woodruff Creek-Urban

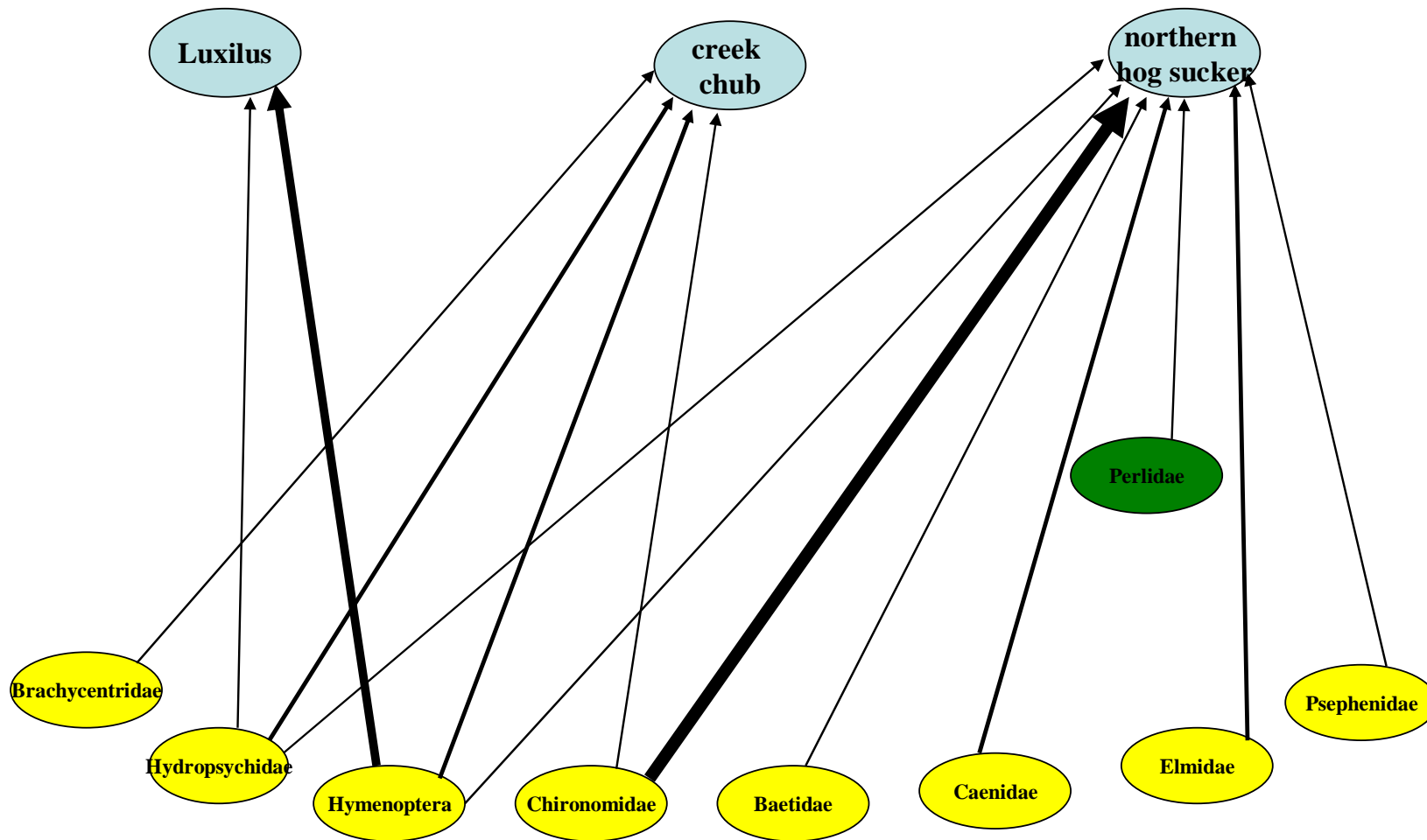


Figure 14. site 164. Mann Creek-Urban

